

Leachate Screening Considerations

PURPOSE: The purpose of this technical note is to present an approach for development of a leachate screening protocol. The protocol will evaluate the acceptability of confined disposal of dredged material and the need to perform laboratory testing and unsaturated and saturated ground-water modeling. This technical note presents the main factors that will be considered during the development of the leachate screening procedures. The impact of these factors on the leachate quality exposed to receptors will be examined, and relationships among the dominant factors and leachate quality will be developed for use in the screening procedures.

BACKGROUND: Contaminated dredged material is often placed in confined disposal facilities (CDFs) designed and operated to control environmental impacts of the disposed material. A CDF is a diked enclosure having structures that retain dredged material solids. CDFs can be upland, nearshore (partially surrounded by water), and in-water (totally surrounded by water) (Figure 1). When contaminated dredged material is placed in a CDF, contaminants may be mobilized to form leachate that is transported to the site boundaries by seepage. Subsurface drainage and seepage through dikes may reach adjacent surface water and groundwater and act as a source of contamination.

Leachate seeping into the groundwater from dredged material placed in a CDF can be produced by several potential sources: gravity drainage of the original pore water and ponded water, inflow of groundwater, and infiltration of rainwater and snowmelt. Thus, leachate generation and transport in a CDF depend on many disposal site-specific and sediment-specific factors. Immediately after dredging and disposal, dredged material is saturated (all voids are filled with water). As evaporation and seepage remove water from the voids, the amount of water stored and available for gravity drainage decreases. Since the contaminants in dredged material are primarily adsorbed to sediment particles, leaching by percolating site water from a CDF situated above the groundwater table is the primary mechanism by which contaminant migration to groundwater takes place. If the site is situated so that groundwater will flow through the material (typically, a nearshore CDF), percolating groundwater may be the primary source of water through the material. If the CDF is a nearshore or island facility, surface water may be in contact with the dredged material as a result of fluctuating water levels and then transport contaminants from the CDF in a process termed tidal or wave flushing.

INTRODUCTION: Section 404 of the Clean Water Act of 1972, as amended, the National Environmental Policy Act of 1969, the U.S. Army Corps of Engineers (USACE) management strategy for dredged material disposal (Francingues et al. 1985), and the USACE/U.S. Environmental Protection Agency (USEPA) technical framework for evaluating the environmental effects of dredged material management alternatives (USACE/USEPA 1992) require the evaluation of the confined disposal alternative for dredged material to include groundwater impacts. Time-varying leachate flow and leachate quality must be predicted to evaluate potential impacts. Because contaminant mobility in dredged materials is variable and highly site specific, laboratory testing

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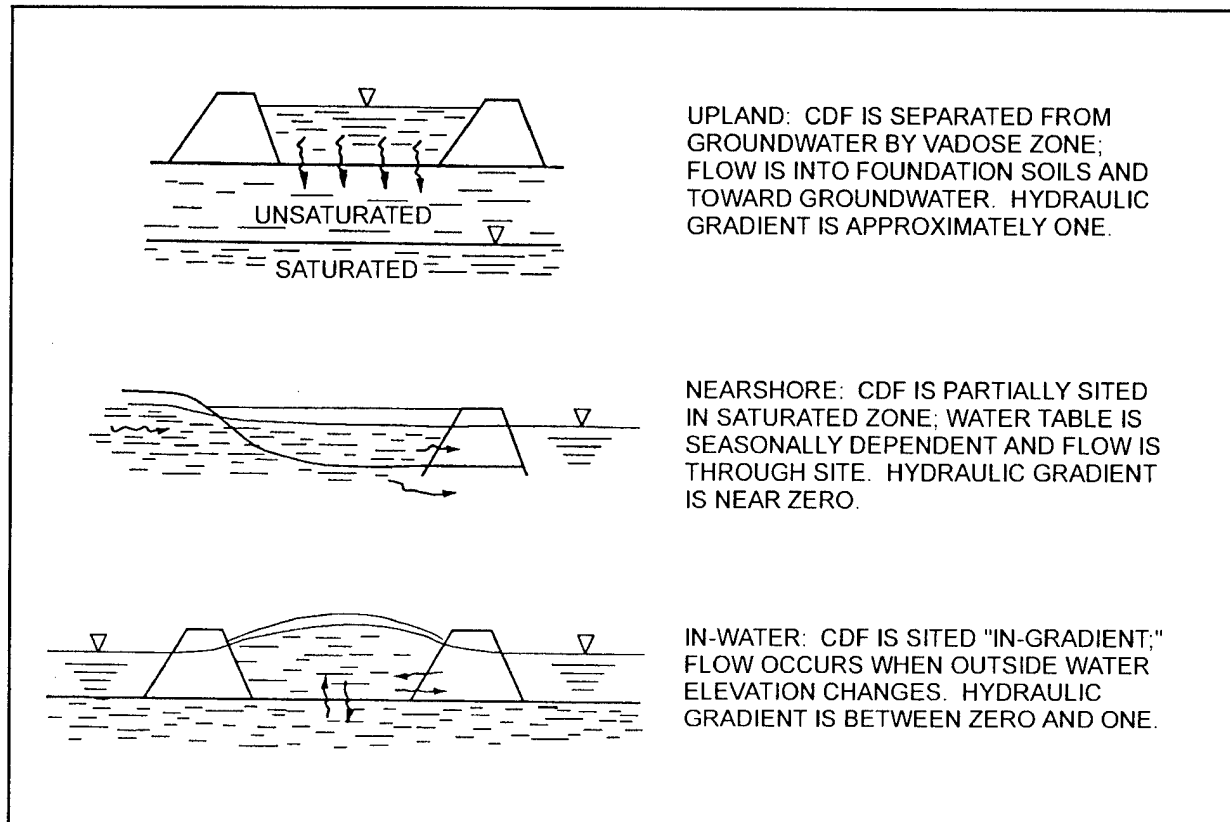


Figure 1. Contaminant migration pathway: leachate seepage

and groundwater modeling have commonly been required to predict leachate quality (Myers and Brannon 1991). Laboratory testing for leachate quality and groundwater modeling can be both expensive and time-consuming and may be unnecessary for all but the most contaminated dredged materials. Conservative screening procedures could be employed to identify scenarios when testing for some or all classes of contaminants would not be needed. This technical note presents the framework for developing such screening procedures.

Theoretical Basis for Leachate Quality Evaluations. Contaminant migration via leachate seepage is a porous medium contaminant transport problem (Figure 2). Leaching is defined as interphase transfer of contaminants from dredged material solids to the pore water surrounding the solids and the subsequent transport of these contaminants by pore water seepage. The interphase transfer is the first step or source of contaminated leachate generation. Interphase mass transfer during dredged material leaching is a complicated interaction of many elementary processes and factors. A complete description of all these processes, factors, and interactions is not presently possible. Instead, a lumped parameter, the distribution coefficient, is used to describe the distribution of contaminant between aqueous and solid phases.

The contaminants in the aqueous phase are convected with pore water in the dredged material as leachate. As leachate is transported through porous media, redistribution of the contaminants between the advected pore water (leachate) and the new solids encountered (the surrounding porous media) occurs, and a new equilibrium between the leachate and the solids is reached. This

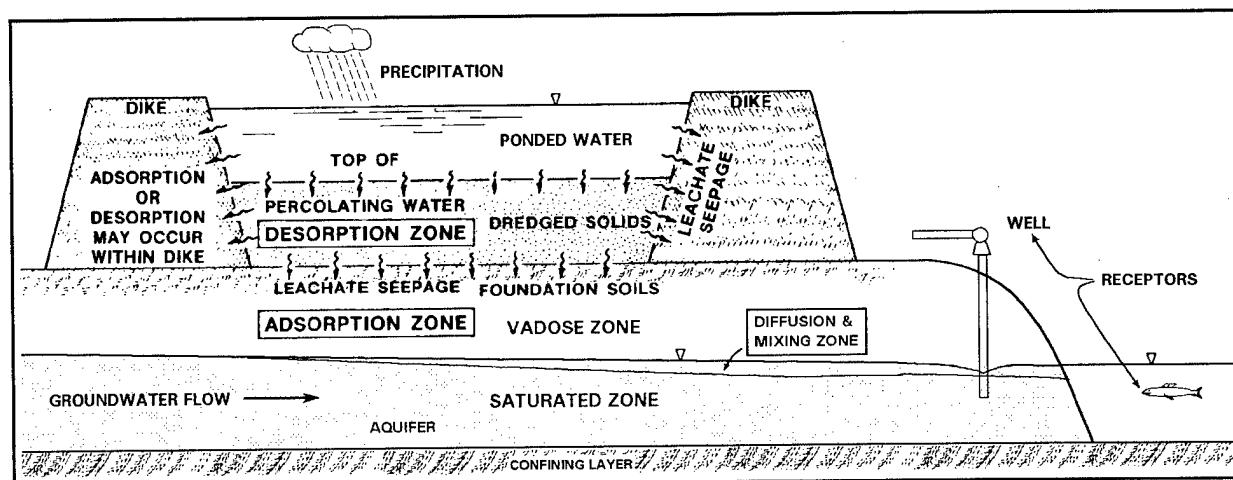


Figure 2. Model of dredged material leaching

redistribution reduces the contaminant concentration in the leachate as it passes through cleaner layers of dredged material, foundation soils, and fine-grained soils in the vadose zone (unsaturated zone).

The contaminant concentration of leachate exposed to a receptor is further impacted by diffusion or mixing as the leachate is transported from the CDF locale to the receptor through the coarse-grained layers of an aquifer. In effect, the contaminant concentration in the leachate is diluted by the groundwater flow. Attenuation by adsorption to organic matter and interactions with fine-grained materials will also occur in the aquifer, but the effect is generally small due to low concentration of organic and clayey materials in the main regions of saturated groundwater flow.

Screening Approach. Based on the theoretical basis of leachate quality evaluations described previously, a three-step screening evaluation is proposed:

- 1) Evaluate the source strength for its potential to generate pore water with unacceptable levels of contaminants and identify the contaminants of concern (those that may pose a leachate problem). Screening values for unacceptable concentrations would be based on the receptors. For example, for human receptors the screening values would be drinking water standards or risk-based values for drinking, bathing, or recreation. Similarly, for aquatic organisms the screening values would be chronic toxicity criteria or risk-based values to protect the food web and populations.
- 2) Evaluate the attenuation by adsorption of contaminants of concern in the CDF, saturated fine-grained foundation soils, and vadose zone and determine which contaminants may pose a leachate problem beyond the CDF locale.
- 3) Evaluate the diffusion (dilution) of the remaining contaminants of concern in the saturated zone between the CDF and the receptor.

These three sequential processes are illustrated in Figure 2 showing leachate transported through the desorption, adsorption, and diffusion zones for an upland CDF. The evaluations are discussed

in detail in the following sections on the source, CDF and vadose zone considerations, and saturated zone and receptor factors. Discussion of the evaluations is focused on upland CDFs but most of the considerations apply as well to all CDFs.

SOURCE: In order for contaminants to cross the interface between dredged material solids and water, a difference in chemical potentials must exist. Chemicals flow from a region of high chemical potential to a region of low chemical potential just as electric current flows from a region of high electrical potential to one of lower electrical potential or as mass flows from a position of high gravitational potential to one of low gravitational potential. When chemical potentials are equal, the net transfer of contaminant across the solid-water interface is zero, and the mass of contaminant in each phase is constant, but not necessarily equal. The processes shown in Figure 3 control the rate at which equilibrium is reached and the equilibrium distribution of contaminant between solid and aqueous phases. Once equilibrium is reached, the ratio of contaminant mass in the solid phase to the contaminant mass in the aqueous phases does not change.

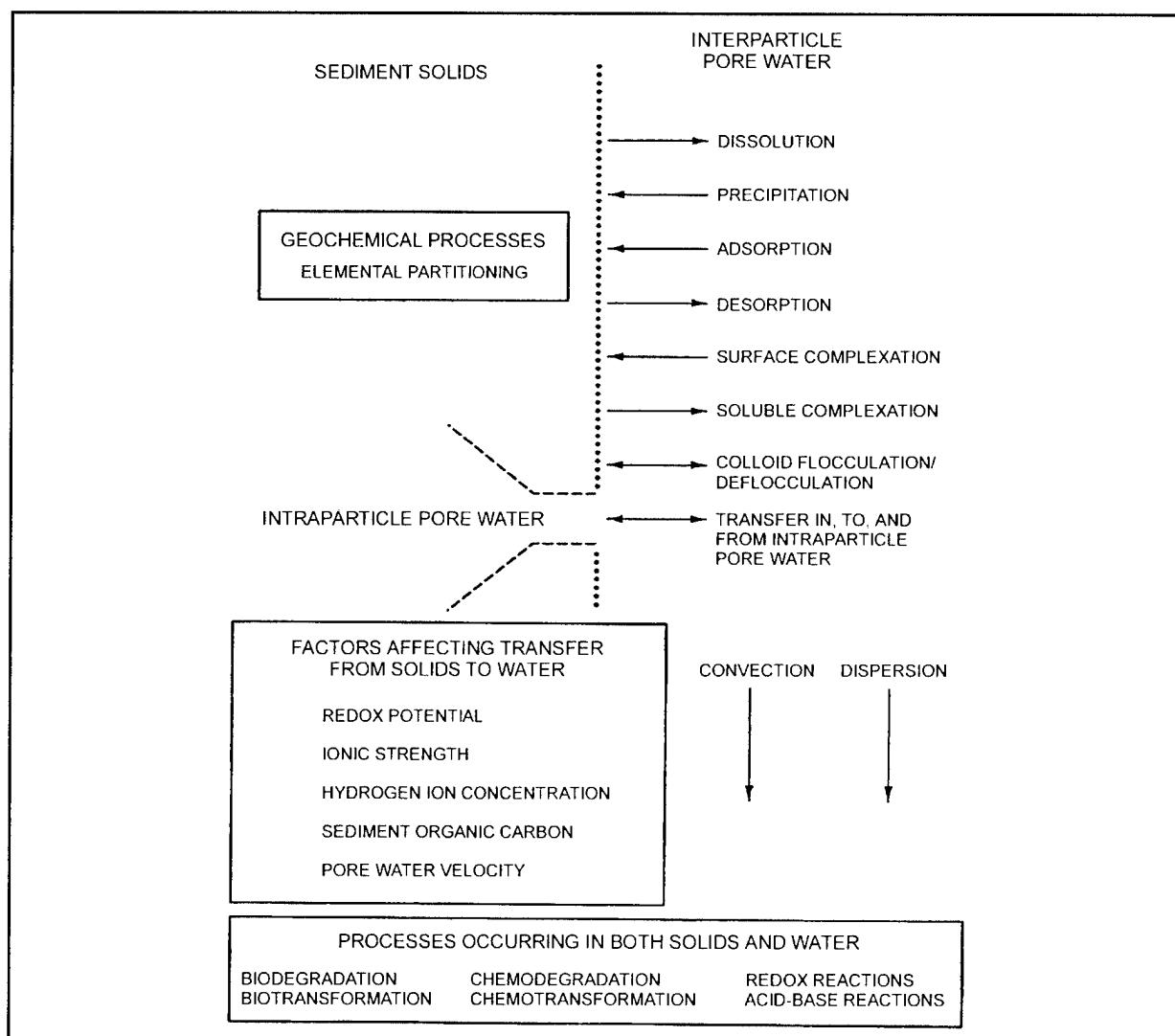


Figure 3. Factors influencing transfer processes between solids and water

Equilibrium Assumption. In practice, a true equilibrium between dredged material solids and pore water never exists because some of the processes shown in Figure 3 have very slow reaction rates. However, a pseudo-steady-state can be reached between dredged material solids and water if the water is moving past the solids slow enough, as discussed in a following section. Assuming equilibrium between solid and aqueous phases eliminates the need for determining controlling processes and the rate coefficients for these processes. Without the equilibrium assumption, laboratory testing and mathematical modeling would require determination of controlling processes and investigation of the kinetics for these processes. As is apparent from Figure 3, predictive laboratory tests and mathematical models based on chemical and mass transfer kinetics would be too complicated for routine application to dredged material leaching. Thus, application of the equilibrium assumption is imperative for the development of predictive techniques suitable for routine use.

Once equilibrium has been reached, only the relative distribution of contaminant between solid and aqueous phases is needed to predict leachate quality.

$$K_d = \frac{q}{C} \quad (1)$$

where

K_d = equilibrium distribution coefficient, L/kg

q = solid-phase contaminant concentration at equilibrium, mg/kg

C = aqueous phase contaminant concentration at equilibrium, mg/L

Equation 1 describes the equilibrium distribution of a single contaminant in a dredged material; that is, equilibrium distribution coefficients are contaminant and dredged material specific. As will be discussed in a later section, K_d is affected by various factors (sediment oxidation status, pH, and ionic strength). Varying these factors during leaching can shift the equilibrium position of the system and change K_d .

Laboratory Testing. Laboratory testing procedures have been developed to quantify the dredged-material-specific equilibrium distribution coefficients considering changes in many of the sediment geochemical factors. A sequential batch leach test (SBLT) has been recommended for leachate testing of freshwater sediments (Brannon, Myers, and Tardy 1994). Because major differences exist in the leaching characteristics of freshwater and estuarine sediments, prediction of leachate quality for estuarine sediments using the SBLT is difficult.

A thin-layer column leach test has been developed to simulate contaminant leaching in CDFs (Myers, Brannon, and Tardy 1996). This test is recommended for leachate testing of estuarine sediments that are dredged and disposed in CDFs for which the primary source of water for leaching is low in ionic strength (e.g., rainfall) relative to the dredged material initial pore water ionic strength. Leaching of estuarine sediments and dredged materials with low-ionic-strength water results in destabilization of the colloidal system as salt is washed out. Colloids are released and, along with the colloids, colloid-bound contaminants.

Equilibrium-Controlled Desorption in a CDF. The assumption of equilibrium-controlled desorption in a CDF is based on two arguments: (a) the intuitive argument that the interphase transfer rates affecting leachate quality are fast relative to the volumetric flux of water in CDFs and (b) the argument that equilibrium-controlled desorption provides conservative predictions of leachate quality. This section discusses these arguments. The term desorption as used here and in the remainder of the technical note refers to the composite effect of the elementary interphase transfer processes (release of contaminants from the solid phase) shown in Figure 3.

Contaminated dredged materials are usually fine grained and have hydraulic conductivities in the range of 10^{-8} to 10^{-5} cm/sec. When the hydraulic conductivity is this low, pore water velocity is also low for the gradients normally encountered in CDFs. Consolidation with excess pore pressure can yield greater localized gradients at the bottom. For gradients near 1, pore water velocities approximate hydraulic conductivities; that is, the water moves very slowly at velocities of 10^{-8} to 10^{-5} cm/sec.

When the rate at which water moves is slow relative to the rate at which equilibrium is approached, a local chemical equilibrium exists between the pore water and the sediment solids. The local equilibrium concept is illustrated in Figure 4. The local equilibrium assumption implies that as a parcel of water passes a parcel of dredged material solids, the water and solids come to chemical equilibrium before the parcel of water moves to contact the next parcel of dredged material solids. Leachate quality at the surface of a CDF will differ from leachate quality at the bottom of a CDF, while leachate in both locations will be in equilibrium with the dredged material solids. In reality, equilibrium-controlled desorption requires an infinitely fast desorption rate. However, if the critical interphase transfer rates are sufficiently fast, the equilibrium assumption can yield results indistinguishable from full kinetic modeling (Jennings and Kirkner 1984; Valocchi 1985; Bahr and Rubin 1987).

In addition to being a good approximation, the assumption of equilibrium-controlled desorption is conservative for desorption; that is, predictions based on the equilibrium assumption will overestimate leachate contaminant concentrations for dredged material. Additionally, the solubility of contaminant in the solids may also further limit the contaminant concentration in the leachate. The equilibrium assumption is conservative because interphase transfer is from the dredged material solids to the pore water. At equilibrium all of the desorption that can occur has occurred. Thus, for clean water entering the dredged material, pore water contaminant concentrations cannot be higher than the equilibrium value. Equilibrium is not a conservative approximation for the adsorption of contaminants in the clean soils below the contaminated dredged material. As such, adsorption is often neglected during transport of leachate in the predominantly coarse-grained materials of an aquifer.

Partitioning Factors

Oxidation status of sediment. Neither hydraulic nor mechanical dredging adds sufficient oxygen to overcome the sediment oxygen demand of polluted sediments. As a result, the dredged material in a CDF is anaerobic except for a surface crust that may develop if the CDF dewateres by evaporation and seepage. Such an oxidized crust may eventually be several feet thick, but seldom represents a significant portion of the vertical profile for the typically fine-grained material in CDFs. An aerobic

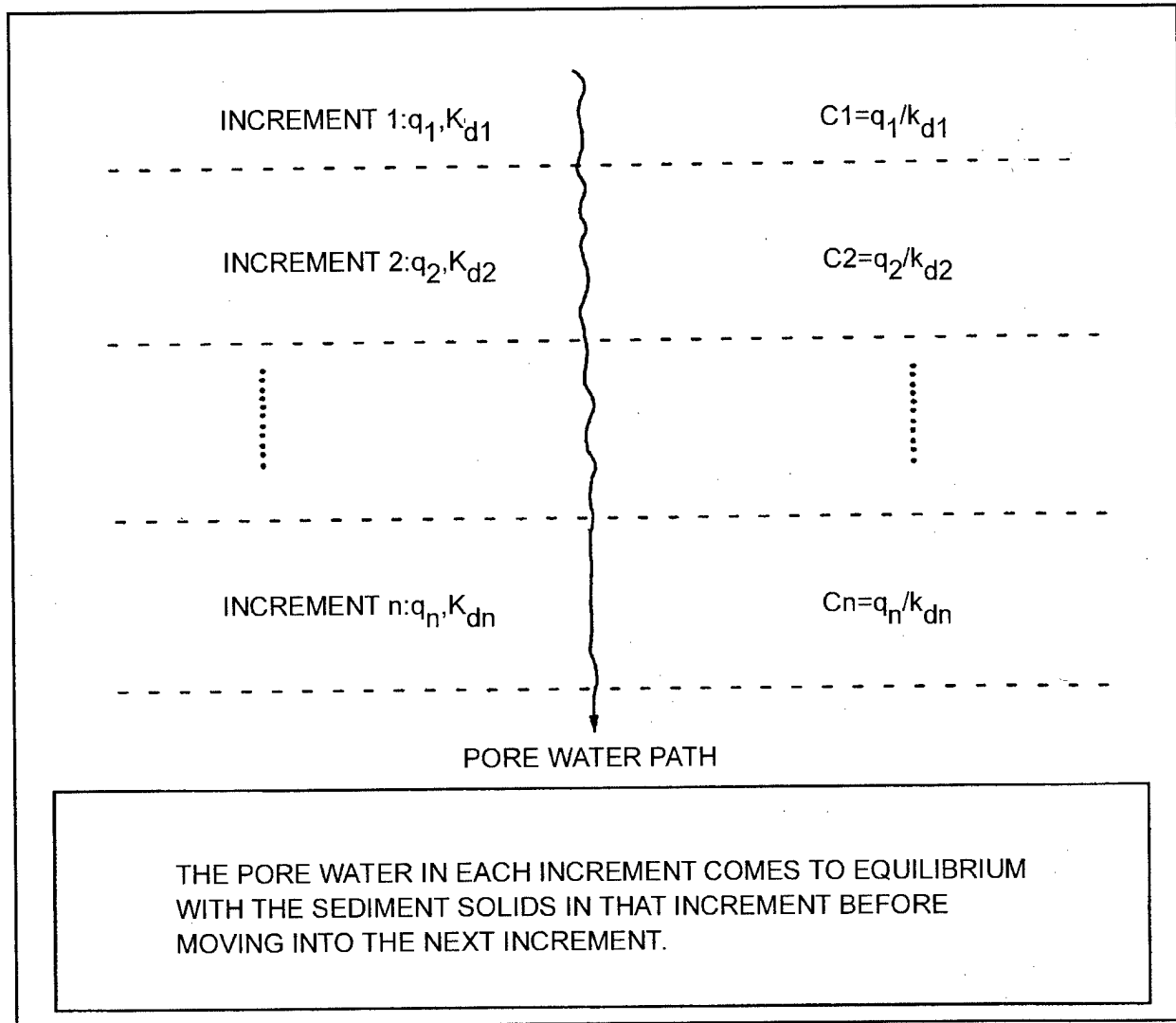


Figure 4. Illustration of local equilibrium assumption in leaching from a CDF

leaching procedure may be necessary if the full lift thickness is dewatered prior to placement of the next lift. Sequential batch leaching of aerobic, aged sediment can be used to simulate leaching of the surface crust in a CDF (Brannon, Myers, and Tardy 1994).

Ionic strength. Sequential batch leaching of freshwater sediments usually yields a desorption isotherm such as shown in Figure 5 (Brannon, Myers, and Tardy 1994). This is what is known as a classical desorption isotherm. Its key feature is a single distribution coefficient that is constant throughout the sequential leaching procedure. A commonly observed feature of desorption isotherms for metals in freshwater sediments is that they do not go through the origin, but intercept the ordinate at some other point. The intercept indicates the amount of metal in geochemical phases that is resistant to aqueous leaching which is generally tied up by acid volatile sulfides in the sediment.

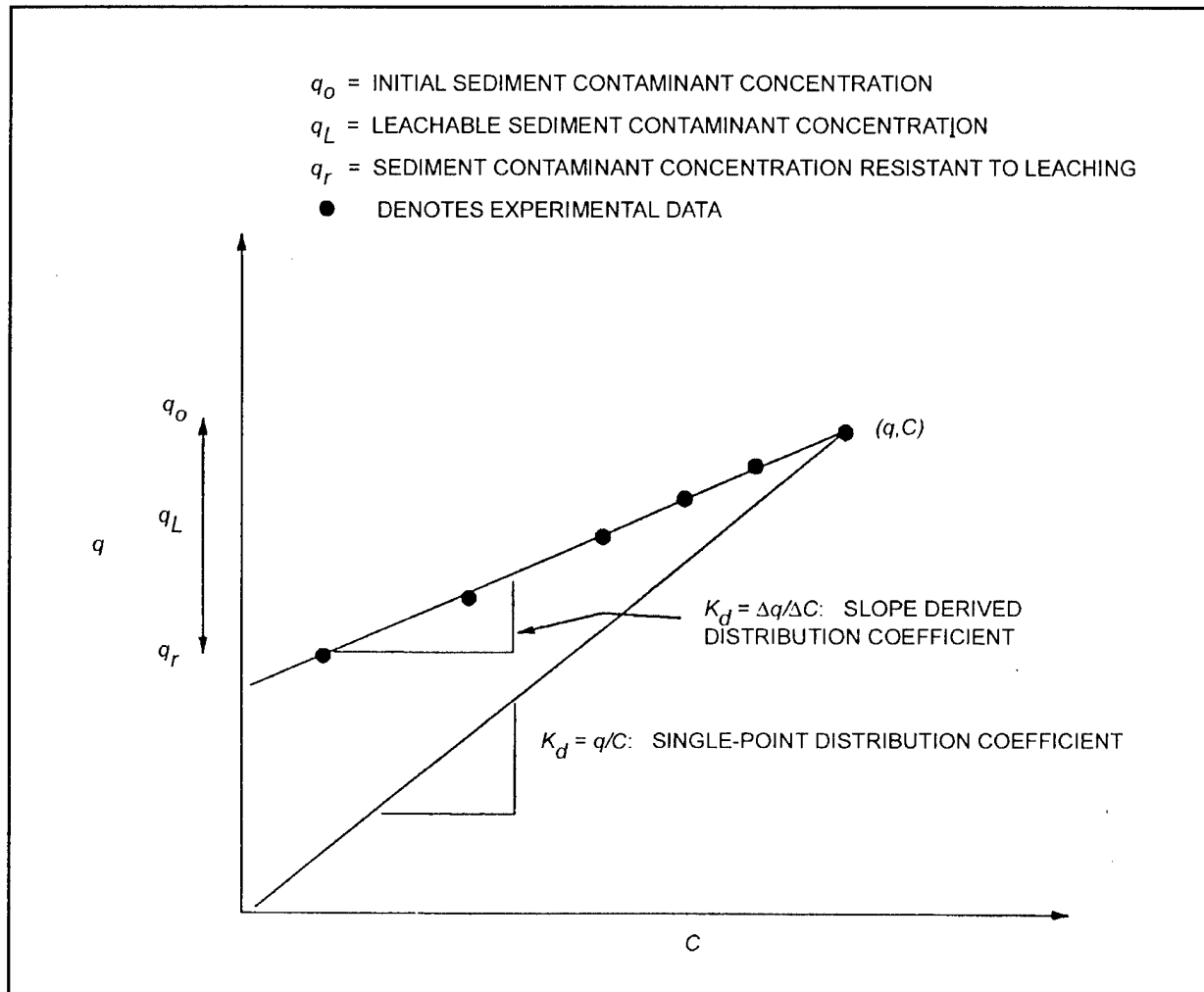


Figure 5. Desorption isotherms for slope-derived and single-point distribution coefficients, typical of freshwater sediments

The general form of the q versus C relationship for classical desorption isotherms is as follows:

$$q = K_d + q_r \quad (2)$$

where q_r is the solid-phase contaminant concentration resistant to leaching, mg/kg.

Nonconstant distribution of contaminants between dredged material solids and water is commonly observed during leaching of estuarine sediments (Brannon et al. 1989; Brannon, Myers, and Price 1990; Brannon et al. 1991). Nonconstant contaminant partitioning yields batch isotherms for which the distribution coefficient changes as the solid-phase concentration q decreases during sequential leaching, until a turning point is reached (Figure 6). At the turning point, the distribution coefficient becomes constant and desorption begins to follow the classical isotherm. The nonconstant distribution coefficient portion of the desorption isotherm is related to elution of salt.

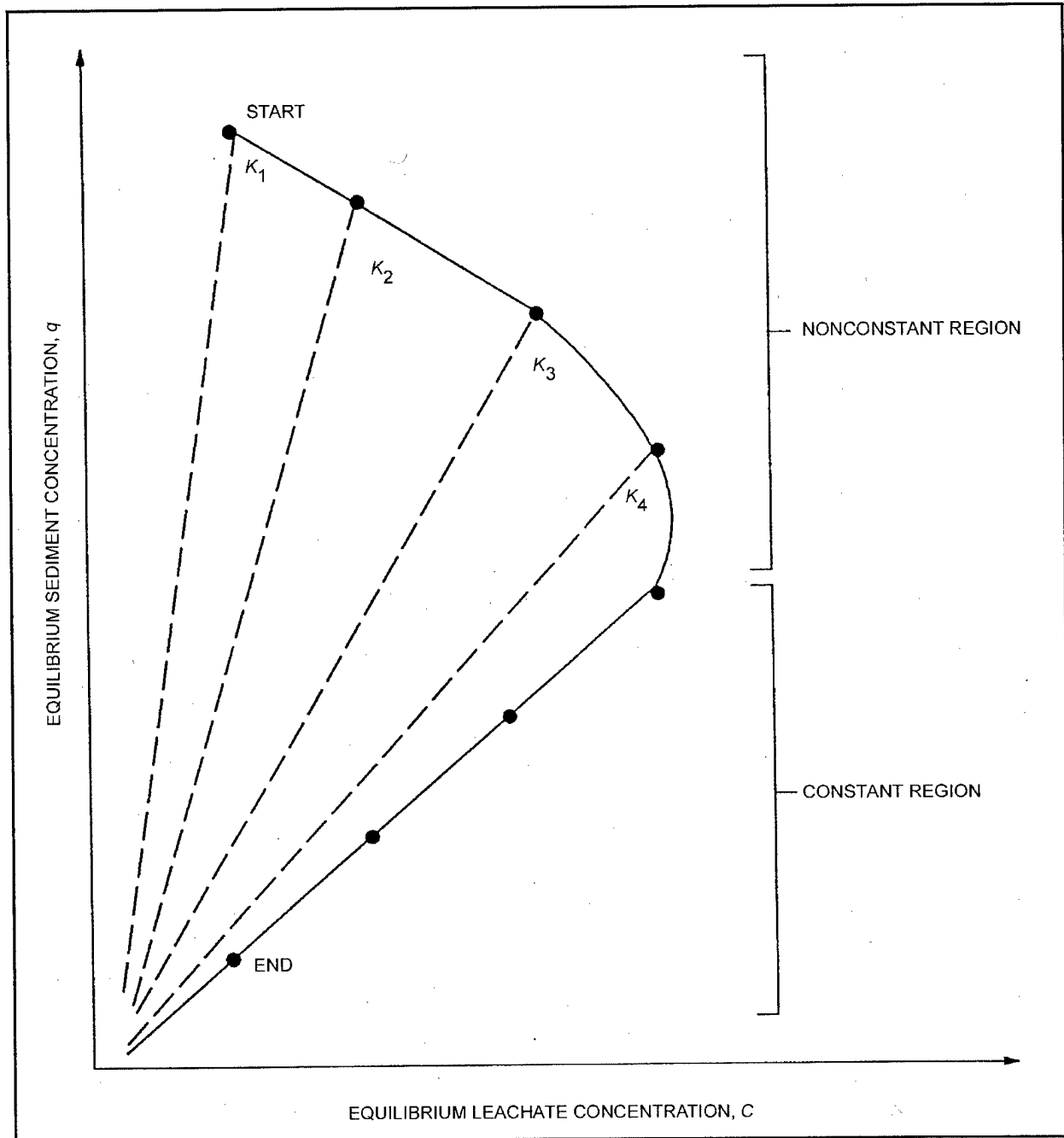


Figure 6. Desorption isotherm illustrating constant and nonconstant partitioning

As salt is eluted from estuarine sediments, the ionic strength of the aqueous phase is reduced. According to the Gouy-Chapman model of charge distribution in double layers, decreasing the ionic strength increases repulsive forces (Stumm and Morgan 1981) and causes the double-layer thickness between colloids to increase. Flocculated colloidal matter becomes increasingly deflocculated and more easily entrained in flow. The overall effect is an increase in dissolved organic carbon (DOC) concentrations in the aqueous phase, mobilizing metals and organic contaminants bound to the colloidal matter (Brannon et al. 1991). For these reasons, the type of desorption isotherm shown in Figure 6 is referred to as a DOC-facilitated desorption isotherm. Since the relationship of q versus

C is not a one-to-one correspondence for DOC facilitated desorption isotherms, q as a function of C cannot be developed from the isotherm.

The shear velocity at particle surfaces affects colloid release from sediment particles under the influence of decreasing ionic strength. The shear velocities developed by agitation during batch testing are infinitely large relative to the low shear velocities developed as water percolates through dredged material in a CDF. Colloidal mass release in a batch test, therefore, is not representative of colloidal mass release in a CDF under the influence of decreasing ionic strength. In addition, batch testing requires a liquid-solids separation step that alters the size distribution of colloids included in the dissolved phase. Thus, in a batch test, neither the mass nor the size distribution of colloidal release to pore waters in a CDF is properly represented. For these reasons, it is difficult to couple results from sequential batch leaching with porous media fluid mechanics (advection and diffusion) and to predict leachate quality without column leaching results.

Source Evaluation. The initial contaminant concentration in the leachate is the contaminant concentration in the pore water of the dredged material. This concentration can be estimated using the partitioning relationships given previously. Under generation of the estimated leachate quality, comparisons between drinking water, surface water, and risk-based standards and the initial contaminant concentrations in the leachate can be made to provide an early indication of potential leachate problems. If appropriate standards are not exceeded for some contaminants, such comparisons complete the leachate evaluation for these contaminants. If the appropriate standards for some contaminants are exceeded, additional evaluations will be needed. Surface water standards should be used only if the leachate is leaving the site and impacting surface waters.

CDF AND VADOSE ZONE CONSIDERATIONS: Two aspects of leachate generation from CDFs are of particular concern: leachate quality and leachate quantity. Leachate generation depends on site-specific hydrology and geohydrology, engineering controls at the disposal site, CDF operation, dredged material hydraulic conductivity, initial water content, and nature of contaminants. Evaluation of potential leachate impacts will be greatly affected by the nature of the site and the engineering controls in place. Varying the engineering controls and site operation during the evaluation also allows selection of the optimum controls.

Transport Factors. CDF siting, design, and operation affect both leachate contaminant concentrations and leachate flow, but predominantly flow. Leachate flow in conjunction with leachate contaminant concentration determines the mass of contaminant that can potentially leave the site boundaries. Contaminant mass leaving site boundaries is particularly important when comparing the effects of various CDF disposal options such as depth of fill, drainage of surface water, and other leachate control measures such as liners, collection, and treatment. The main CDF factors affecting leachate generation are climate, siting and foundation properties, dredging and disposal methods, CDF design and operation, disposal sequence, and control features.

Climate. Climate influences the infiltration of precipitation into the CDF and the evapotranspiration from materials in the CDF. Greater precipitation increases the potential infiltration and leachate generation. Greater temperature, solar radiation, and wind and lower humidity decrease the potential evapotranspiration and leachate generation. The distribution of precipitation throughout the year also affects the potential infiltration and evapotranspiration. Higher precipitation during

winter months when potential evapotranspiration is lowest produces conditions for greater infiltration and leachate generation. Precipitation from large, intense storms produces greater runoff and therefore less infiltration and leachate than gentle rainfall for precipitation events of equal magnitude. Precipitation on frozen soil also produces greater runoff and therefore less infiltration and leachate than precipitation on unfrozen soil.

Siting. Several siting factors influence the leachate evaluation. Among the more important factors are foundation properties, foundation thickness, location of receptors, and geohydrology. Foundation soils that are in a reduced state and have high pH, organic matter, mineral oxides, and acid volatile sulfides retard contaminant mobility by increasing contaminant retention. Foundation soils with low permeability restrict leachate flow. These properties are more common in fine-grained soils. Thicker foundations of fine-grained soils provide greater retention of contaminants. The location of receptors and geohydrology are important because greater distance from the CDF and the path of leachate flow reduces the contaminant concentration exposed to the receptor. Similarly, siting a CDF for saltwater dredged material over a saltwater aquifer reduces the potential for contaminating a freshwater aquifer. Areas with high groundwater velocities provide greater dilution of the leachate plume, but spread the leachate plume more quickly.

Dredging and disposal methods. Hydraulic dredging or disposal as opposed to mechanical dredging and disposal greatly increases the initial water content of the dredged material, providing a greater short-term source of leachate. As such, the required storage volume for a given quantity of in situ sediment is much larger, requiring a larger CDF in depth or area. Increasing the area increases the leachate volumetric flow rate. Increasing the depth of dredged material increases the pressure head driving leachate production as can be seen in Figure 1, especially for an upland CDF. Additionally, hydraulic operations greatly increase the short-term hydraulic conductivity of the dredged material, increasing the rate of leachate production. In the long term (after several years) the leachate flux for the hydraulically dredged or placed material will approach the flux for mechanically dredged and placed material as the material consolidates from the dewatering.

Hydraulic dredging or disposal also separates the material into a mound of predominantly coarse-grained material and a layer of fine-grained material. This process serves to concentrate the contaminants in the layer of fine-grained material. This may change both the leachate flow rate and quality in the short and long term. Leaching from the sand mound may increase the leachate production due to its low contaminant retardation and high permeability; however, the concentration of contaminants in the sand mound would be expected to be much lower than in the fine-grained layer.

CDF design and operation. CDF design and operation can affect both leachate quantity and quality. Leachate quantity increases with the area of the CDF. Leachate quantity may decrease with increasing dewatering efforts and promotion of runoff. Dewatering and consolidation of the dredged material decrease the pressure head driving drainage through the CDF and decrease the hydraulic conductivity of the dredged material, both serving to decrease leachate production.

Desiccation of the dredged material will cause volatile and semivolatile organic contaminants to volatilize, reducing their concentrations in the leachate. If the entire thickness of dredged material in the CDF is fully desiccated, the material will become oxidized and the pH may drop if the sediment

is high in acid volatile sulfides. Oxidation and pH reduction increase the concentration of a number of metals of concern in the leachate. Additionally, oxidation increases the quantity (leachable fraction) of the metals in the dredged material that can leach.

Disposal sequence. Sandwiching the placement of the most contaminated dredged material between layers of less contaminated material or placing clean fine-grained material at the bottom of the CDF provides a sorptive capacity within the CDF to retard the contaminant migration and to improve the leachate quality. This type of layering and encapsulating greatly improves the initial leachate quality, and the improvement may last for decades. In addition, the layering serves effectively to homogenize or equalize the dredged material quality in terms of the long-term leachate quality, significantly reducing the maximum contaminant concentrations in the leachate.

Control features. Liners and drains are the primary control features for leachate. Liners can greatly restrict leachate flow rates from CDFs. Liners also divert leachate to drains that collect the leachate and route it to a treatment facility. The control measures prevent nearly all of the leachate from reaching any of the receptors.

Evaluation of CDF and Vadose Zone Effects. To evaluate CDF and vadose zone effects on leachate concentration, site-specific considerations of factors affecting leachate generation must be considered. After dredging and disposal, dredged material is initially saturated (all voids are filled with water). As evaporation and seepage remove water from the voids, the amount of water stored and available for gravity drainage decreases. After some time, usually several years for conventional CDF designs, a quasi-equilibrium is reached in which water that seeps or evaporates is replenished by infiltration through the surface. The amount of water stored when a quasi-equilibrium is reached and the amount released before a quasi-equilibrium is reached depend primarily on local hydrology, dredged material properties, and facility design factors. To predict time-varying leachate flow, all of these factors must be considered.

Preproject estimation of leachate flow, therefore, requires coupled simulation of local weather patterns and hydrologic processes governing leachate generation. Important climatic processes and factors include precipitation, temperature, solar radiation, wind speed, and humidity. Important hydrologic processes include infiltration, snowmelt, runoff, and evaporation. Important subsurface processes include evaporation from dredged material voids and flow in unsaturated and saturated zones. The Hydrologic Evaluation of Leachate Production and Quality (HELPQ) model (Aziz and Schroeder 1999) can be used to simulate these processes for selected disposal scenarios.

Quantifying the CDF and vadose zone effects on leachate concentration from its source to the saturated zone for a wide range of the dominant CDF factors including interactions among the factors provides a basis for evaluating the leachate at the point of entry into the saturated zone without running the HELPQ model. This quantification is anticipated in the development of the screening procedure. The HELPQ model could be run for the site-specific conditions if increased accuracy in the predictions were needed to pass the screening.

SATURATED ZONE AND RECEPTOR FACTORS: Output from the HELPQ model giving the leachate flow rate and contaminant concentration from the bottom of the CDF or vadose zone as a function of time can be linked to a multimedia model, such as the Multimedia Environmental

Pollutant Assessment System (Streile et al. 1996), or a groundwater model, such as the DOD Groundwater Modeling System (Environmental Modeling Research Laboratory 1999), to allow for prediction of the contaminant concentrations at the receptors. The saturated zone model will predict the diffusion of the leachate and the adsorption of contaminants on fine-grained and organic matter. Important factors in the modeling are the groundwater velocity, the location of the receptors, the thickness of the aquifer, the heterogeneity of the aquifer, and the quantity of fines and organic matter in the aquifer. Only those contaminants having a concentration that exceeds its water quality standard upon entering the saturated zone need to be evaluated with a saturated zone model.

Transport Considerations

Groundwater velocity. Groundwater velocity increases the diffusion and dilution of the leachate plume, but decreases the time that it takes for the leachate plume to reach a receptor.

Receptor locations. Receptors that are located where the groundwater table is upgradient from the CDF or significantly off the center line of the leachate plume have a small chance of exposure to the leachate plume. Additionally, the leachate concentration would be greatly diluted by diffusion into the groundwater.

Aquifer thickness. Aquifers with greater thicknesses have more water in which the leachate can diffuse. Greater thickness has the potential to significantly decrease the leachate concentration exposed to distant receptors.

Aquifer heterogeneity. Aquifer heterogeneity has the potential to increase short-circuiting of the leachate flow to a receptor. Short-circuiting would decrease diffusion and dilution of the leachate plume.

Retardation capacity. The capacity of an aquifer to retard the contaminants in the aquifer is a function of the quantity of fine-grained materials, oxides, sulfides, and organic matter in the aquifer media. If contaminants are present in significant quantities, the long-term exposure will be greatly reduced. Most aquifers show little retardation, and the retardation tends to be a short-term phenomenon.

Evaluation of Transport in Saturated Zone. To evaluate transport effects on leachate concentration in the saturated zone, site-specific aquifer properties affecting leachate diffusion and contaminant retardation must be considered. Leachate seeps from the CDF and moves through the vadose zone to the saturated zone. The leachate is then transported as a plume in the groundwater. The plume diffuses and entrains groundwater, becoming more dilute as it moves downgradient with the groundwater. Quantifying the dilution and retardation along the center line of the plume as a function of distance and time from the entry into the saturated zone for a wide range of aquifer properties provides a basis for evaluating the leachate at the receptor without running a groundwater model. Additional refinement of the quantification of the dispersion as a function of distance perpendicular to the center line can improve the estimate of contaminant concentrations at points off the center line. This quantification is anticipated in the development of the screening procedure. A groundwater model could be run for the site-specific conditions if increased accuracy in the predictions were needed to pass the screening.

CONCLUSIONS: A leachate screening protocol is being developed that considers the entire leachate pathway from the dredged material (source) through the CDF, foundation (vadose zone), and the saturated zone to the receptors. The protocol is based on the theoretical considerations presented in this technical note. The approach taken consists of three steps:

- 1) Evaluate the source strength for its potential to generate pore water with unacceptable levels of contaminants and identify the contaminants of concern (those that may pose a leachate problem) based on equilibrium partitioning between the dredged material solids and the pore water. Screening values for unacceptable concentrations would be based on the receptors.
- 2) Evaluate the attenuation by adsorption/retardation of contaminants of concern in the CDF and vadose zone and determine which contaminants may pose a leachate problem beyond the vadose zone. For screening, relationships among the controlling factors in the CDF and vadose zone and the attenuation of leachate concentration will be developed using the HELPQ model.
- 3) Evaluate the diffusion (dilution) of the remaining contaminants of concern in the saturated zone between the CDF and the receptor. For screening, relationships among the controlling factors in the saturated zone and the attenuation of leachate concentration will be developed using the MEPAS model and the GMS.

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